CSIRO PUBLISHING

www.publish.csiro.au/journals/ijwf *International Journal of Wildland Fire* **2008**, *17*, 463–475

Assessing the differenced Normalized Burn Ratio's ability to map burn severity in the boreal forest and tundra ecosystems of Alaska's national parks

Jennifer L. Allen[A,C] *and Brian Sorbel*[B]

[A]National Park Service, Fairbanks Administrative Office, 4175 Geist Road, Fairbanks, AK 99709, USA.

[B]National Park Service, Alaska Regional Office, 240 W 5th Avenue, Anchorage, AK 99501, USA.

[C]Corresponding author. Email: jennifer_allen@nps.gov

Abstract. Burn severity strongly influences post-fire vegetation succession, soil erosion, and wildlife populations in the fire-adapted boreal forest and tundra ecosystems of Alaska. Therefore, satellite-derived maps of burn severity in the remote Alaskan landscape are a useful tool in both fire and resource management practices. To assess satellite-derived measures of burn severity in Alaska we calculated the Normalized Burn Ratio (NBR) from pre- and post-fire Landsat TM/ETM+ data. We established 289 composite burn index (CBI) plots in or near four national park areas between 2001 and 2003 in order to compare ground-based measurements of burn severity with satellite-derived values of burn severity. Within the diverse vegetation types measured, a strong linear relationship between a differenced Normalized Burn Ratio (dNBR) and CBI for eight out of the nine fire assessments was found; R^2 values ranged from 0.45 to 0.88. The variations in severity among four pre-fire vegetation types were examined and a significant difference in the average dNBR and average CBI values among the vegetation types was found. Black spruce forests overall had the strongest relationship with dNBR, while the high severity white spruce forests had the poorest fit with dNBR. Deciduous forests and tall shrub plots had the lowest average remotely sensed burn severity (dNBR), but not the lowest ground severity among the vegetation types sampled. The tundra vegetation sampled had the lowest ground severity. Finally, a significant difference was detected between initial and extended assessments of dNBR in tundra vegetation types. The results indicated that the dNBR can be used as an effective means to map burn severity in boreal forest and tundra ecosystems for the climatic conditions and fire types that occurred in our study sites.

Additional keywords: dNBR, Landsat, *Picea*, remote sensing, wildland fire.

Introduction

Wildland fires are an integral part of the boreal forest and tundra ecosystems of Alaska; at least 20 million hectares (ha) have burned in the past 55 years (Kasischke *et al*. 2002; Bureau of Land Management 2006). Alaskan wildland fires burned more than 5.6 million ha between 2001 and 2006, which accounted for 34% of the total area burned by wildland fire throughout the United States (National Interagency Coordination Center 2007). The total area burned annually in Alaska varies greatly (Kasischke *et al*. 2002) but in nearly all years some area of Alaska is affected by large wildland fires. For instance, during every Alaskan fire season since 1999 at least one fire has grown larger than 45 000 ha (Bureau of Land Management 2006).

Climate, terrain and vegetation strongly influence the occurrence, extent and severity of fires in Alaska. On heterogeneous landscapes fires nearly always burn in a non-uniform manner. As fires burn under varying weather conditions across landscapes characterised by varying topography and fuel types, their behaviour and effects also dramatically change (Johnson 1992; Johnson and Miyanishi 2001). Thus, within any given fire, some areas are radically changed as a result of intense scorching or sustained burning of surface organic layers, while other areas remain untouched. This heterogeneous pattern or 'fire mosaic' is the result of varying burn severity on the landscape. Although numerous definitions of burn severity exist in the literature (Lentile *et al*. 2006), we define burn severity as a measure of the ecological impacts of fire, in terms of plant survivorship or mortality, depth of the burn in organic layers, or amount of biomass consumed.

Burn severity can shape vegetation succession and the patch mosaics on the landscape (Johnstone and Chapin 2006). The patchy dynamics of burn severity can result in substantial heterogeneity at both a local and landscape level. Studies have shown that burn severity can impact both tree seedling establishment (Zasada 1986; Johnstone and Kasischke 2005; Johnstone and Chapin 2006) and understorey regeneration strategies (Zasada 1986) in boreal forests. These variations at the local level can be scaled up to impact landscape scale patch dynamics (Johnstone and Chapin 2006).

Conversely, vegetation can also influence fire behaviour and resulting severity. For instance, it is commonly thought that in boreal forests deciduous stands are less flammable than

 10.1071/WF08034 1049-8001/08/040463

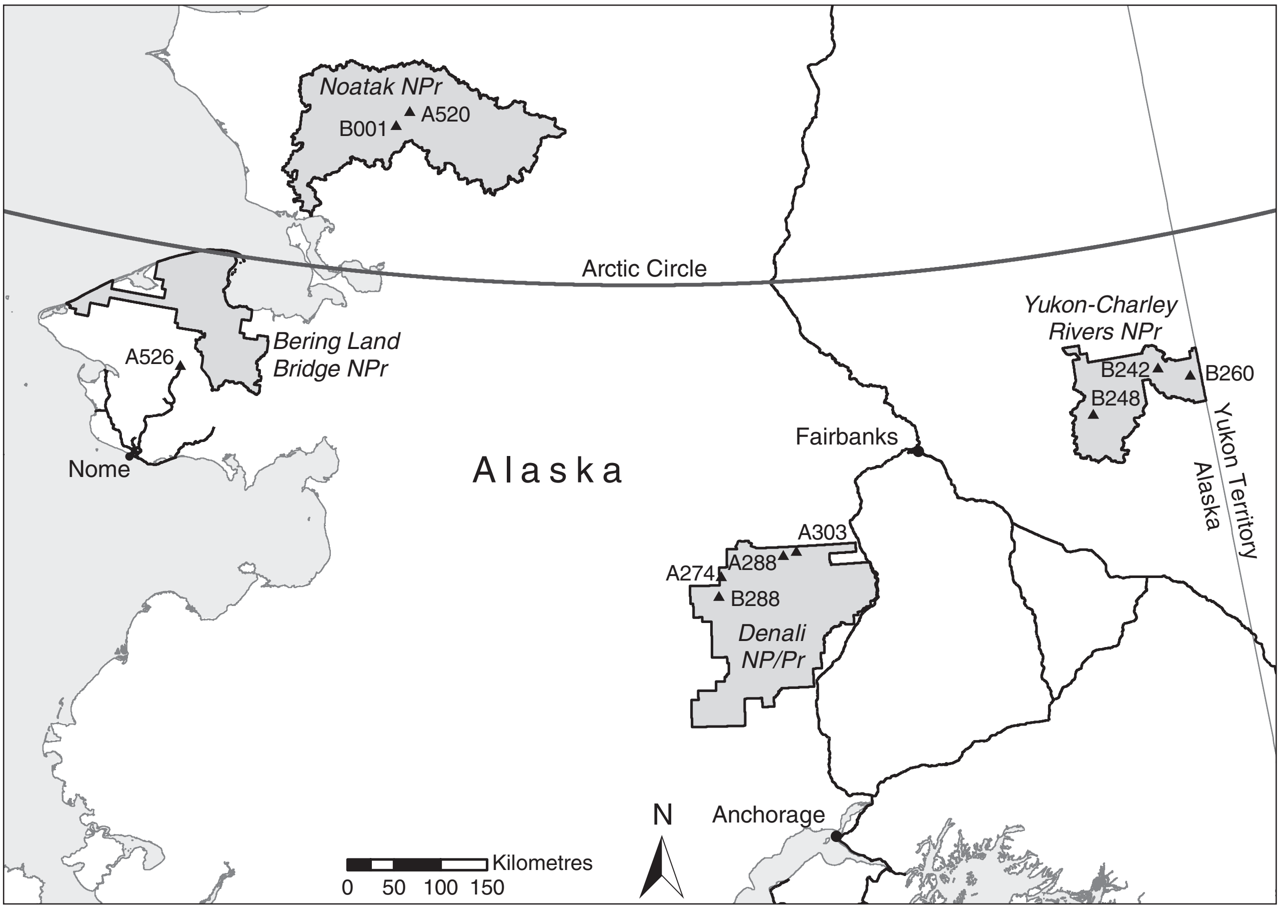

Fig. 1. Location of 10 fires (▲) assessed for burn severity within and near Alaska Region National Park Service units.

mature spruce stands (Johnson 1992). Yet very few studies have documented the variation in burn severity (or fire behaviour) within some vegetation types in Alaska, such as tussock tundra, deciduous forests, and white spruce forests.

Information about burn severity has several practical applications for fire and resource managers. For example, the United States Interagency Burned Area Emergency Response teams (BAER) often create and use burn severity maps to direct management responses to minimise the potentially harmful effects of excessive erosion in the aftermath of a wildland fire (Miller and Yool 2002). Furthermore, useful predictive models of wildland fire effects on vegetation communities and wildlife populations can be generated from burn severity products. Effective resource and fire management practices require an accurate, inexpensive and timely means to map the burn severity of wildland fires.

Several approaches to mapping burn severity of wildland fires have been evaluated recently (Epting *et al*. 2005; Lentile *et al*. 2006; Roy *et al*. 2006; French *et al*. 2008). The Normalized Burn Ratio (NBR) has emerged as one means to map burn severity using remote sensing technology (Key and Benson 2006). The NBR utilises the differing responses of Landsat bands 4 (near infrared, 0.76–0.90 μm) and 7 (mid infrared, 2.08–2.35 μm) to generate a scaled index of burn severity. A differenced Normalized Burn Ratio (dNBR) dataset compares the NBR of pre- and post-fire Landsat imagery to isolate the change brought about by wildland fire. The application of the dNBR to Landsat imagery has shown to be an effective method to map burn severity within temperate forests (van Wagtendonk *et al*. 2004; Cocke *et al*. 2005) and boreal forests (Epting *et al*. 2005), although other studies have suggested that the inconsistent response of the bands used in dNBR make it an unreliable method (Roy *et al*. 2006). Two types of burn severity assessments can be produced: initial assessments or extended assessments (Key and Benson 2006), however no studies have compared the differences between these assessment types within boreal forest or tundra ecosystems.

In this study, we evaluated the ability of the dNBR to accurately map the burn severity of wildland fires within boreal forest and tundra ecosystems in Alaska by comparing remotely sensed dNBR burn severity values with ground-based burn severity measurements. We assessed the burn severity of 10 fires that occurred between 1999 and 2002 in or near four National Park Service (NPS) units in Alaska: Yukon–Charley Rivers National Preserve (YUCH), Denali National Park and Preserve (DENA), Noatak National Preserve (NOAT) and Bering Land Bridge National Preserve (Seward Peninsula) (Fig. 1, Table 1). These

Table 1. Fires assessed for burn severity within the Alaska Region of the National Park Service

NPS unit	Fire name	Fire number	Fire year	Fire size (ha)	Field plots
YUCH	Witch	B242	1999	19 002	32
YUCH	Jessica	B260	1999	15 448	47
YUCH	Beverly	B248	1999	8160	40
DENA	Otter Creek	A288	2000	4685	10
DENA	Chitsia	A303	2000	3776	25
DENA	Foraker	A274	2000	7267	24
DENA	Herron River	B288	2001	2524	25
NOAT	Cottonwood Bar	A520	2002	5486	19
NOAT	Uyon Lakes	B001	2002	174	18
Seward peninsula	Milepost 85	A526	2002	8720	49

fires occurred over a wide geographic area of Alaska, and provided the ability to evaluate the dNBR and ground-based severity in both the boreal forest types of Interior Alaska, as well as the tussock-shrub tundra communities of north-western Alaska, thus extending the previous research of Epting *et al.* (2005). We were interested in determining whether dNBR values had the same meaning among different vegetation types and between assessment types. We compared the relationship of ground and remotely sensed burn severity among vegetation types and dNBR assessment types. While the results from three YUCH fires have been previously presented by Epting *et al.* (2005), we include these data to provide a comprehensive comparison among all fires and vegetation types assessed on national park lands in Alaska.

Methods

Study areas

Yukon–Charley Rivers National Preserve is a 1.02 million ha preserve located along the Canadian border in central Alaska (Fig. 1). Elevations range from 150 to 1950 m. The Preserve includes ~200 km of the Yukon River and the entire Charley River drainage. It is characterised by boreal forests that transition from river floodplains through steep mountain slopes to alpine tundra. Black spruce (*Picea mariana* (P. Mill.) B.S.P.) forests generally occupy north-facing, low lying, or poorly drained sites. Well drained upland forests are dominated by white spruce (*Picea glauca* (Moench) Voss) and mixed forests of white spruce and paper birch (*Betula papyrifera* Marsh.) or quaking aspen (*Populous tremuloides* Michx.). White spruce and balsam poplar (*Populus balsamifera* L.) are common along streams and flood plains. The Yukon–Charley area is characterised by one of the highest lightning strike densities in Alaska (Dissing and Verbyla 2003). Vegetative and climatic factors at Yukon–Charley enable wildland fires to ignite and burn; over 40% of the preserve has burned within the past 50 years. Fires B242, B248 and B260 occurred in Yukon–Charley Rivers NP in the summer of 1999. All three fires occurred in mountainous terrain at elevations ranging from 200 to 1200 m, with the majority of the fire areas below the tree line (~900 m).

Denali National Park and Preserve covers 2.43 million ha in central Interior Alaska. Elevations range from 60 m in the southern portion of the park to 6194 m at the summit of Mt. McKinley. Wildland fire activity in Denali primarily occurs in the north-western portion of the park where lightning-caused fires predominate, and along the road corridor in the eastern portions of the park where human-caused fires predominate. The north-west region of Denali lies within the interior lightening belt of Alaska with lightning strike densities only slightly lower than those in Yukon–Charley (Dissing and Verbyla 2003). The lowland black spruce forests that are common throughout the north-west region of the park experience large natural fires almost yearly. Over the past 50 years nearly 220 000 ha have burned within the park boundary. Fires A274, A288 and A303 occurred in Denali in 2000. Fire B288 occurred in 2001. All four fires occurred in the north-west region of the park and ranged over elevations from 200 to 500 m. The primary pre-fire vegetation in this area was woodland/stunted black spruce, shrub tussock tundra and some white spruce forests in flood plains and riparian zones.

Noatak National Preserve encompasses 2.66 million ha in north-western Alaska. Elevations in the preserve range from roughly sea-level to 1900 m. Fires generally occur in the lowlands of the park, below 500-m elevation. The Noatak River is one of the major river systems of Arctic Alaska and contains a broad range of lowland and alpine tundra, including tussock tundra, shrub tundra and tree-line habitats. Fires A520 and B001 occurred in the central portion of the preserve along the Noatak River in 2002. The pre-fire vegetation in the areas burned by these two fires was dominated by tussock-shrub tundra, birch-ericaceous shrub tundra and willow shrublands. The two fires burned at elevations that ranged from 180 to 430 m.

Bering Land Bridge National Preserve is a 1.13 million ha preserve located in north-western Alaska on the Seward Peninsula. In 2002, fire A526 burned near the western edge of the preserve in areas representative of the tussock-shrub tundra communities within the preserve. Portions of the fire had previously burned in 1997. Fire A526 burned at elevations ranging from 50 to 340 m. The Seward Peninsula has one of the lowest lightning strike densities in Interior Alaska (Dissing and Verbyla 2003). Fire suppression activities did not impact the natural progression of any of the fires in this study.

Image processing

Burn severity GIS data layers were developed for all 10 fires by applying the NBR to pre- and post-fire Landsat imagery based on the Landscape Assessment methods developed by Key and Benson (2006). All image processing for the 10 fires was conducted by the USGS EROS Data Center as part of the NPS-USGS Burn Severity Mapping Project. For each fire, suitable cloud-free pre-fire and post-fire scenes were selected from nearly the same time of year to minimise differences in vegetation phenology between scenes. The selected images were georeferenced to the terrain correction level (Level 1T) using ground control points and a Digital Elevation Model for topographic accuracy, and processed to units of at satellite reflectance. Neither an atmospheric correction nor scene-to-scene radiometric normalisation was applied to the Landsat images used in this analysis. A NBR dataset was

Table 2. Fires and satellite imagery used to develop differenced normalised burn severity products

Fire number	Fire year	Pre-fire scene[A]		Post-fire scene		Assessment type
B242, B260	1999	L5 (p66/r14)	16 September 1995	L7 (p65/r14)	12 September 1999	Initial
B248	1999	L5 (p66/r15)	16 September 1995	L7 (p67/r14)	10 September 1999	Initial
A288, A303	2000	L5 (p70/r15)	1 July 1986	L7 (p70/r15)	16 June 2001	Extended
A274	2000	L5 (p71/r15)	21 July 1985	L7 (p71/r15)	23 June 2001	Extended
B288	2001	L7 (p71/r16)	28 August 1999	L7 (p71/r16)	26 August 2001	Initial
B288	2001	L7 (p72/r15)	23 June 2001	L7 (p72/r15)	17 June 2002	Extended
B001	2002	L7 (p78/r12)	8 July 2001	L7 (p78/r12)	29 July 2002	Initial
A520	2002	L7 (p80/r12	8 August 2001	L7 (p79/r12)	8 August 2003	Extended
A526	2002	L7 (p81/r14)	3 August 2002	L7 (p81/r14)	6 August 2003	Extended

[A]L5: Landsat 5, L7: Landsat 7.

generated for both the pre-fire and post-fire scene using Landsat bands 4 and 7 as follows:

$$\text{NBR} = 1000[(\text{Band 4} - \text{Band 7})/(\text{Band 4} + \text{Band 7})]$$

A final dNBR dataset was derived as follows:

$$\text{dNBR} = \text{NBR}_{\text{pre-fire}} - \text{NBR}_{\text{post-fire}}$$

The dNBR has a potential range of -2000 to $+2000$, but rarely exceeds a range of -500 to $+1200$ over burned surfaces (Key 2006; Key and Benson 2006). Unchanged areas typically have values near zero, indicating no change, and burned areas generally have dNBR values greater than 100, although this burned/unburned threshold often varies between burns in different regions. A national study of over 80 fires or fire complexes in the USA found the range of burned/unburned thresholds to be between 80 and 130 (Zhu *et al*. 2006). The dNBR is hypothesised to generate a continuous index of severity, from unburned to severely burned.

Key and Benson (2006) described two general types of burn severity assessments: initial assessments and extended assessments. The assessment types differ in the timing of the Landsat imagery used with respect to the fire event. To generate the initial assessment dNBR, the post-fire Landsat scene is acquired immediately after fire activity has ceased or the fire has been declared out. In contrast, with extended assessments the post-fire Landsat scene is typically acquired one growing season after the fire event. Both assessments then use a pre-fire scene acquired on or near the anniversary date of the post-fire scene. Because of cloud and smoke issues with imagery, we were only able to obtain both an initial and extended assessment for fire B288 in Denali. For the other nine fires, either initial or extended assessments were completed. Table 2 summarises the imagery used and assessment type for each of the 10 fires in the analysis. Ideally, there should only be one to two years separating the pre-fire and post-fire imagery to minimise the change detected in unburned areas that was not the result of the subject fire. However, because of scene limitations, it was sometimes necessary to have longer periods of time between the two scenes (Table 2). In the case of fire A274 in Denali, there was a difference of 16 years between the pre-fire and post-fire scenes used in the analysis. We do not expect that this longer time span between images in Denali significantly impacted our results, because of the slow changes in stand characteristics in late successional boreal forest types (Viereck 1983).

Composite burn index plots

We used the composite burn index (CBI) to estimate burn severity in the field (Key and Benson 2006). A total of 289 CBI plots were established within the 10 fires (Table 1). We collected field data from 119 burn severity plots within the three Yukon–Charley fires between June and July 2001 (two years after the fires occurred). In June 2002, we collected 84 burn severity plots within the four Denali fires (one and two years after the fire). While the CBI protocol generally calls for sampling plots during the first post-fire growing season (Key and Benson 2006), we do not feel that sampling two years post-fire negatively impacted our ability to determine burn severity in these predominantly forested systems. In July 2003 (one year post-fire), we collected 47 burn severity plots from fire A526 near Bering Land Bridge and 36 plots from two fires in Noatak. Of the 47 plots from fire A526, 10 were in areas that had previously burned in 1997. At these plots, an attempt was made to separate the burn severity effects of the 2002 burn from the effects of the 1997 burn. Plot locations were pre-selected and stratified to ensure sampling of the full range of burn severity levels and vegetation types within the fires. In addition, plot locations were targeted for areas with little spatial heterogeneity in burn severity. This was done to reduce the potential for plots to be located on edges of burn severity types or in areas with vastly divergent dNBR values from one cell to the next. To find areas with relatively similar dNBR values in neighbouring cells, we used a neighbourhood GIS function to determine the range of dNBR values in a 3×3 matrix around each cell. Plot locations were targeted for those cells in which the range of dNBR values in a 3×3 matrix around the cell was less than 150. All CBI plots were located at least 100 m away from each other. Landsat imagery was not available at the time of field sampling for the Herron River fire; therefore, plots were established based on aerial reconnaissance for homogeneous burn severity areas.

Plots were circular with a 15-m radius in forested plots and a 10-m radius in shrub/tundra plots. For each plot we made ocular estimates of the degree of environmental change caused by fire within five strata defined as: (1) substrate layer, (2) low vegetation less than 1 m tall, (3) tall shrubs/sapling trees (1–2 m), (4) intermediate trees (2–8 m), and (5) large trees

Table 3. Composite Burn Index strata and measures for 2001–2003

Strata	2001 variables	2002 variables	2003 variables
Substrate layer	–	Litter	Litter/light fuel consumed
	–	–	Duff
	Moss[A]	Moss[A]	Moss[A]
	<1000-h fuels	<1000-h fuels	Tussocks[A]
	≥1000-h fuels	≥1000-h fuels	
	Soil cover/colour	Soil cover/colour	Soil cover/colour
Herbaceous/low shrub	–	Nonvascular plants	% foliage altered
	Regeneration	% living/resprouting	% living/resprouting
	New serals	New serals	Colonizers
	δ richness/cover	Species diversity	Species composition – relative abundance
Tall shrubs/sapling trees	% consumed	% foliage consumed	% foliage altered
	–	% conifers green	% green
	Re-gen shrub/trees	% living/resprouting	% living/resprouting
	New serals	New serals	–
	δ richness/cover	Species diversity	Species composition – relative abundance
Intermediate trees	% green	% green	Not present in tundra plots
	% black	% black	
	% brown	% brown	
	Char height	Char height	
	% girdled	–	
Large trees	% green	% green	Not present in tundra plots
	% black	% black	
	% brown	% brown	
	Char height	Char height	
	% girdled	–	

[A]Variables that used Alaska specific definitions or treatment.

(>8 m) (Table 3). Within each stratum four to five variables were scored between 0.0 and 3.0 for a scale of burn severity (Key and Benson 2006). Between 2001 and 2003, the FIREMON Landscape Assessment CBI field form was revised on an annual basis. As a result, the specific variables scored varied slightly from year to year and are shown in Table 3. In addition, slight modifications were made to the CBI field form each year to reflect specific Alaska vegetation conditions. If a particular component was not present at a plot, no score was recorded. The individual factor scores were averaged to provide a continuous scale of severity between 0.0 and 3.0 for the five strata and overall CBI. Plot locations were established using Garmin GPS units, and averaged at least 100 individual points. We used the plot location data to extract the dNBR values from the 30-m cell in which each plot fell. Digital photos and general vegetation descriptions of the plots were taken. Using field notes and digital photographs, one of the following four vegetation classes was assigned to each plot: (1) white spruce (included pure white spruce, mixed spruce, and white spruce-deciduous stands), (2) black spruce (black spruce and black spruce-deciduous), (3) deciduous (deciduous trees and tall shrubs), and (4) tundra (tussock, shrub-tussock, and dwarf or low shrub). We generally followed the vegetation classification definitions commonly used in Alaska (Viereck *et al*. 1992). Plots were classified as forest if there was greater than 10% canopy cover of tree species or tall shrubs in the plot area. Deciduous stands were differentiated from mixed conifer–deciduous plots if greater than 75% of the tree canopy was dominated by deciduous species. Four plots were excluded from the vegetation analyses because they were rocky, dry bluff communities, or alpine plant dominated and did not conform to one of the four classes.

Analysis

In order to assess the ability of dNBR to predict on the ground burn severity we utilised linear regression. We initially examined each fire individually, except in cases where fires occurred in the same year and shared the same pre-fire and post-fire scenes. In those cases, data from the fires were combined for the purpose of analyses (see Table 2), which reduced the number of fires from 10 to 8. For fire B288 which had both initial and extended assessments completed we used the initial assessment dataset for all analyses unless otherwise noted. We used analysis of covariance (ANCOVA) to determine whether the linear regression relationships varied among the eight fires. If there was a significant difference, we ran a second ANCOVA to test the significance of the interaction term between the two independent variables to determine if there was a significant difference among the slopes. Pearson correlation coefficients (R) were generated to compare the five individual strata of CBI to dNBR to determine if certain stratum had higher correlations to the dNBR values.

Table 4. Results of linear regressions for CBI as a factor of dNBR by fire and Landsat scene

Assessment type	NPS unit	Fire (by Landsat scene)	Slope	Adjusted R^2	F-statistic (d.f.)	P value
Initial assessment	NOAT	B001	0.0016	0.784	62.68 (1.16)	<0.0001
	YUCH	B242_B260	0.0019	0.748	231.97 (1.77)	<0.0001
	YUCH	B248	0.0018	0.450	32.88 (1.38)	<0.0001
	DENA	B288	0.0024	0.875	169.72 (1.23)	<0.0001
Extended assessment	DENA	B288[A]	0.0025	0.832	119.61 (1.23)	<0.0001
	DENA	A274	0.0024	0.752	70.62 (1.22)	<0.0001
	DENA	A288_A303	0.0019	0.789	127.94 (1.33)	<0.0001
	NOAT	A520	0.0025	0.777	63.86 (1.17)	<0.0001
	SEWARD	A526	0.0022	0.770	161.98 (1.47)	<0.0001

[A]B288 fire had both initial and extended assessments completed, for all subsequent analyses the initial assessment data was used, unless noted.

We examined the effects of pre-fire vegetation (white spruce, black spruce, deciduous, and tundra) on burn severity and then compared the relationships between CBI and dNBR. Using the ground plots, we compared the overall difference in ground (CBI) and remotely sensed burn severity (dNBR) among the pre-fire vegetation types using a general linear model, one-way analysis of variance (GLM ANOVA). We then compared the relationship between CBI and dNBR within each vegetation type using linear regressions and looked for differences using an ANCOVA. Since we were interested in detecting differences in burn severity among vegetation types we chose not to apply the Relative dNBR which others have used to account for differences in vegetation cover, particularly in sparse vegetation communities (Zhu *et al.* 2006; Miller and Thode 2007).

In order to determine if there was an overall difference because of the dNBR assessment types, we pooled the data for all the fires by assessment types (initial *v.* extended) and compared the regressions using an ANCOVA. Since we had both initial and extended assessments completed for fire B288, we analysed these separately. Finally, we pooled the data by pre-fire vegetation and assessment type and compared the resulting regressions. For this analysis, we only utilised black spruce and tundra plots, because of the low number of plots in the other vegetation types when the data was split by assessment type. We used the SPSS 13.0 statistical package for all analyses. The significance level of statistical tests was set at $P < 0.05$.

Results

Within our study we found that ground-based burn severity (CBI) had a significant linear relationship with remotely sensed dNBR values for all fires (Table 4, Figs 2 and 3). Most of the fires had a strong relationship; the adjusted R^2 values were equal to or greater than 0.75 for eight out of the nine assessments. Fire B288 in Denali had the best fit, while the Yukon–Charley fire B248 had the poorest fit (Table 4). The intercepts were significantly different ($F = 8.107$, $P < 0.0001$, Table 5) among the fires; however, the interaction term (Fire × dNBR) indicated that the slopes were not significantly different ($F = 1.295$, $P = 0.253$).

Comparing the five strata of CBI values (Table 6), we found that the intermediate tree strata had the strongest correlation

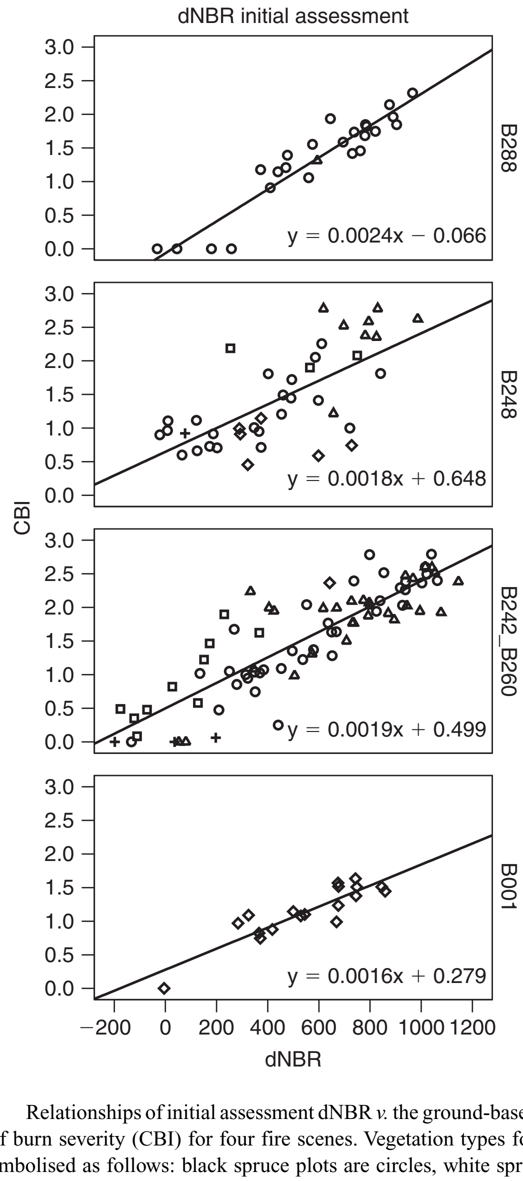

Fig. 2. Relationships of initial assessment dNBR *v.* the ground-based measure of burn severity (CBI) for four fire scenes. Vegetation types for plots are symbolised as follows: black spruce plots are circles, white spruce are triangles, deciduous are squares, tundra are diamonds, and other vegetation types are crosses.

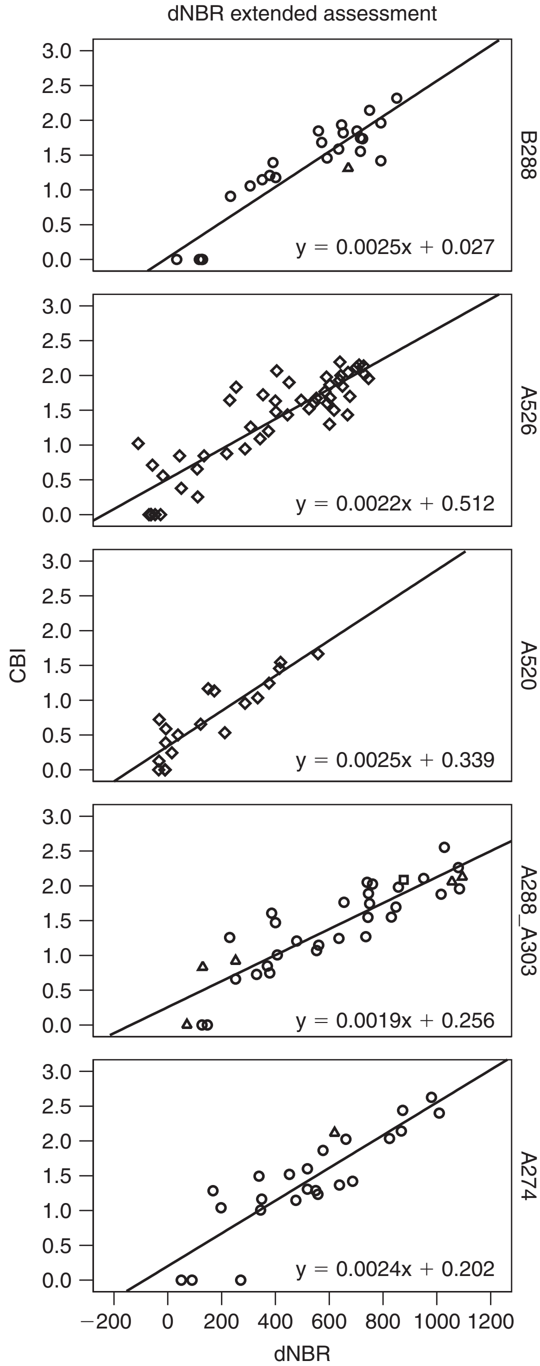

Fig. 3. Relationships of extended assessment dNBR *v.* the ground-based burn severity (CBI) for five fires, including B288 extended assessment. See Fig. 2 for symbol legend.

to dNBR ($r = 0.832$, pooled for all fires), followed by the substrate layer ($r = 0.741$). Total CBI (all strata averaged) had only a slightly stronger relationship ($r = 0.833$ for pooled data) than the intermediate tree strata.

The pre-fire vegetation varied among the parks and fires. In Denali 91% of the 84 plots were woodland (10–25% canopy) and open (25–60% canopy) black spruce, the remaining plots

Table 5. ANCOVA results comparing dNBR as a continuous factor of CBI grouped by fires

Source	Sum of squares (Type III)	d.f.	Mean square	*F*	*P*
Corrected model	110.134[A]	8	13.767	102.642	<0.0001
Intercept	9.064	1	9.064	67.582	<0.0001
dNBR (covariate)	98.474	1	98.474	734.200	<0.0001
Fire (fixed)	7.611	7	1.087	8.107	<0.0001
Error	37.555	280	0.134		
Total	695.908	289			
Corrected total	147.689	288			

[A]Adjusted $R^2 = 0.738$.

were 8% white spruce and 1% deciduous (trees and tall shrubs). All of the 86 plots from Noatak and the Seward Peninsula were classified as tundra, although this class encompassed a wide variety of tundra types including tussock-shrub, low shrub, dry acidic lichen-shrub, and graminoid-dominated tundra. Yukon–Charley had the most diverse vegetation sampled. Of the 119 plots 49% were black spruce, 31% white spruce, 11% deciduous, 6% tundra and 3% other vegetation.

Pooling the plots into pre-fire vegetation types, we found a significant difference in the average dNBR ($F = 18.69$, $P < 0.0001$) and average CBI values among the vegetation types ($F = 11.27, P < 0.0001$) (Fig. 4). The ground-based severity continued to hold a linear relationship with dNBR when separated by pre-fire vegetation (Table 7, Fig. 5). The linear regression relationships varied by vegetation. We found a significant difference among the intercepts ($F = 6.413$, $P < 0.0001$) (Table 8), but the interaction term (Veg × dNBR) indicated that the slopes were not significantly different ($F = 0.138$, $P = 0.937$). Overall, black spruce plots had the strongest relationship between dNBR and CBI ($R^2 = 0.72$). Overall white spruce stands had the highest average dNBR and CBI values of the plots sampled, and also the poorest fit with CBI ($R^2 = 0.58$). Deciduous forest and tall shrub types (deciduous), although not abundantly represented ($n = 14$), had the lowest average dNBR (217.1, s.e. 86.9), followed by tundra with an average dNBR of 378.1 (s.e. 28.4). However, tundra plots had lower average CBI values (average 1.15, s.e. 0.07) than deciduous plots (1.23, s.e. 0.20). We found that the deciduous and tundra plots, when pooled by vegetation type, did not have as strong of correlation between dNBR and CBI as was found with black spruce plots (Table 7).

When we initially compared the data by dNBR assessment types irrespective of fire or dominant pre-fire vegetation type, we found no significant difference in the relationship between dNBR and CBI between initial and extended assessment types ($F = 0.158$, $P = 0.652$; Fig. 6). This was further assessed by comparing fire B288 which had both an initial and extended assessment completed. The R^2 results were similar between the assessment types for B288, with an adjusted R^2 of 0.88 for the initial and 0.83 for the extended assessment (Table 4). There was no significant difference between the relationships for the two assessment types of fire B288 ($F = 0.333, P = 0.567$). However, we found that when we pooled the data by pre-fire vegetation

Table 6. Correlations between the remotely sensed dNBR and field-based burn severity (CBI) strata and CBI total
Correlations are shown for each fire and for all fires pooled. Significance at the 0.05 level is shown with an asterisk (two-tail)

Fires	Pearson correlation	Substrate CBI	Low shrub CBI	Tall shrub/ saplings CBI	Intermediate trees CBI	Big trees CBI	Total CBI
A274	*R*	0.847*	0.767*	0.470*	0.787*	1.000*	0.873*
	n	24	22	22	22	2	24
A288_A303	*R*	0.889*	0.755*	0.698*	0.855*	0.440	0.892*
	n	35	32	32	31	5	35
A520	*R*	0.887*	0.870*	–	–	–	0.889*
	n	19	19	0	0	0	19
A526	*R*	0.860*	0.874*	0.911*	–	–	0.880*
	n	49	49	9	0	0	49
B001	*R*	0.852*	0.939*	0.357	–	–	0.893*
	n	18	18	4	0	0	18
B242_B260	*R*	0.770*	0.676*	0.692*	0.878*	0.920*	0.866*
	n	79	78	78	78	8	79
B248	*R*	0.691*	0.641*	0.466*	0.628*	−0.665	0.681*
	n	40	40	39	36	4	40
B288	*R*	0.900*	0.852*	0.812*	0.915*	–	0.938*
	n	25	25	25	25	0	25
All fires pooled	*R*	0.741*	0.676*	0.656*	0.832*	0.733*	0.833*
	n	289	283	209	192	19	289

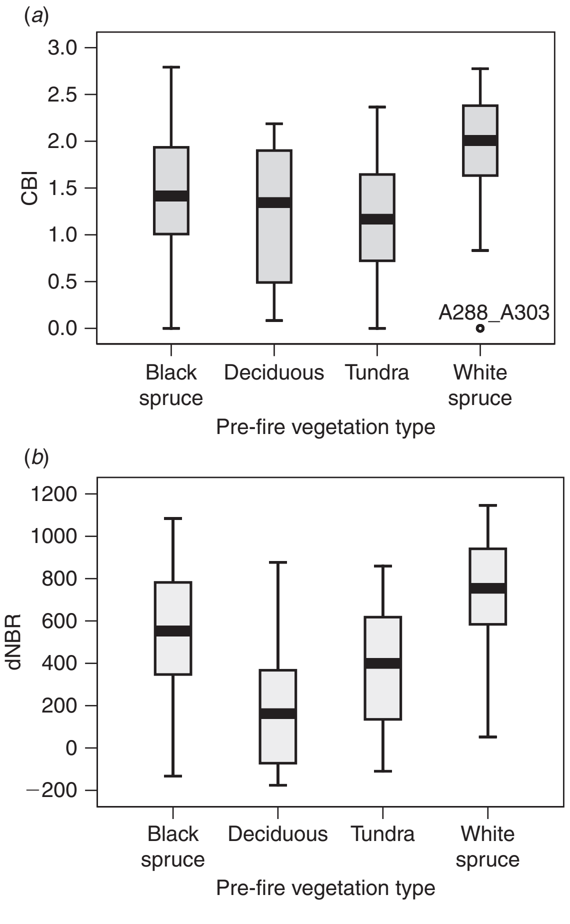

Fig. 4. Box-plots of CBI and dNBR showing the median, quartiles, and extreme values within each vegetation type. Black spruce, $n = 134$; white spruce, $n = 44$; tundra, $n = 99$; deciduous, $n = 14$.

and assessment type, the tundra plots showed a significant difference between the assessment types ($F = 38.07$, $P \leq 0.0001$, $n = 25$ initial assessment (IA) plots, $n = 68$ extended assessment (EA) plots; Fig. 7), while there was no significant difference in the black spruce assessment types ($F = 2.53$, $P = 0.114$, $n = 82$ IA, $n = 52$ EA). Within tundra vegetation, the extended assessment regression had a significantly steeper slope than the initial assessment regression ($F = 6.308$, $P = 0.014$), which indicated that dNBR initial assessments were higher than extended assessment values for the same level of ground severity. The fit of the regression also varied among the two assessment types of tundra (Table 9).

Discussion

The ability to quantify fire impacts is important for fire management, research and monitoring. We found that burn severity values generated from remotely sensed data (dNBR) were highly correlated with ground measurements of burn severity (CBI) for nearly all fires in our study. This finding was similar to other studies (van Wagtendonk *et al.* 2004; Cocke *et al.* 2005; Epting *et al.* 2005), although contrary to studies in Alaska that focussed on fire events from the summer of 2004 (Hoy *et al.* 2008; Murphy *et al.* 2008). Our results indicated that the regression equations and strength of the correlation between dNBR and CBI varied among fires (Table 4) and vegetation community types (Table 7). For instance, we found that dNBR was a good indicator of CBI in black spruce forests but to a lesser degree in white spruce forests.

Black spruce is the dominant boreal tree species in interior Alaska and is often regarded as having severe, stand replacing fires; while white spruce forests are discussed in the literature as being less flammable (Kasischke *et al.* 2006). A study from Canada suggests white spruce stands are less likely to burn than black spruce stands (Cumming 2001). Yet, very few studies

Table 7. Results of linear regressions for CBI as a factor of dNBR when plots are pooled by pre-fire vegetation types

Pre-fire vegetation type	Number of plots	Slope	*y*-intercept	Adjusted R^2	*F*-statistic (d.f.)	*P* value
Black spruce	134	0.00195	0.344	0.722	347.07 (1132)	<0.0001
White spruce	44	0.00186	0.550	0.583	61.078 (1.42)	<0.0001
Tundra	93	0.00184	0.460	0.628	156.38 (1.91)	<0.0001
Deciduous	14	0.0019	0.813	0.685	29.28 (1.12)	<0.0001

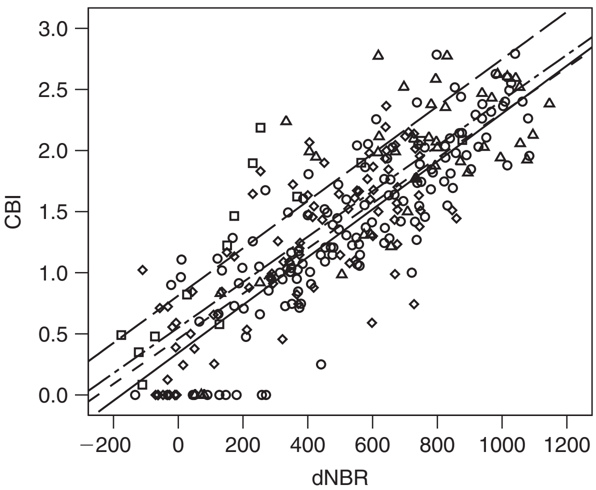

Fig. 5. Relationship of dNBR *v.* CBI grouped by pre-fire vegetation types. Vegetation types for plots are symbolised as follows: black spruce, circles and solid line; white spruce, triangles and dash/dot line; deciduous, squares and long dash line; and tundra, diamonds and short dash line.

have documented the burn severity of white spruce stands on the ground (but see Wang 2002). Our results suggest that white spruce and mixed-white spruce upland forests have the potential to burn just as severely as black spruce. We found that white spruce stands had the highest average severity values for both CBI and dNBR. Most of these plots were in Yukon–Charley (37 out of 44 plots), occurred on steep slopes, and had >25% canopy cover (not shown); a combination often related to high intensity fires within conifer forests, regardless of species (Johnson 1992).

We found that CBI values from black spruce forests had the best overall fit with dNBR ($R^2 = 0.72$), while the CBI values from the higher severity white spruce forests we sampled had the poorest fit ($R^2 = 0.58$). The poor fit may be attributed to the potential for dNBR to level off at high severities. For instance, van Wagtendonk *et al*. (2004) found an asymptote in the relationship that occurred around 750 dNBR or ~2.5 CBI. Our results also indicate a leveling off of the correlation between CBI and dNBR at higher severity levels. The poor fit may also be a result of the narrow range in burn severity, with a majority of the plots having CBI scores greater than one (on a scale of 0 to 3) (Fig. 4).

Based on fire behaviour modelling, we expect low severity surface fires to occur within deciduous forest types (Van Wagner 1983; Johnson 1992). Studies in Canada that compared mixed-wood forests found that fire behaviour models predicted lower intensity fires in forests with a greater deciduous component (Hely *et al*. 2000) and a ground-based study showed that burn severity increased with conifer density (Wang 2002). An Alaskan study that was based entirely on remote sensing data also reported lower burn severity in deciduous compared with coniferous forests on flat terrain (Duffy *et al*. 2007). Within our study, the deciduous vegetation types had the lowest average dNBR values, compared with the other vegetation classes. However, we found that the overall CBI scores were not lowest for deciduous forests (Fig. 4). This suggests that in deciduous forests dNBR values underestimate severity on the ground and, therefore, a different scale of severity should be used. However, we are not sure whether this result is specific to our study, since we had a small sample size (14 deciduous plots) and late timing of scene imagery collection. Most of the deciduous plots were from YUCH. The late scene imagery (mid-September) used for the YUCH fires may have resulted in lower dNBR values. Leaf-off appears to have occurred in the pre-fire scene (data not shown), which could have caused an underestimation of dNBR values or altered detection of dNBR in deciduous stands.

The tussock tundra and low shrub-tussock tundra plots we sampled had the lowest ground severity overall (Fig. 4). Although few studies have documented severity in tundra ecosystems, this is not surprising because of the rapid fire spread (less smouldering) and the generally mesic sites where tussock tundra occurs. However, we did sample some high severity sites where the tussock cottongrass (*Eriophorum vaginatum* L.) was killed off by the fire. Overall the tundra fires from the north-west (B001, A526 and A520) had strong correlations between dNBR and CBI when analysed by fire (Table 4). The fit of the regression, however, was poorer when data was pooled across fires (tundra $R^2 = 0.63$, Table 7).

In comparing dNBR assessment types our results suggest there may be no difference between initial and extended assessments within black spruce forests, but a significant difference within the tundra vegetation class (Fig. 7), where the extended assessments had significantly steeper slopes than initial assessments. This indicates that for the same level of ground severity, initial assessments will have higher values of dNBR than extended assessments in tundra ecosystems. This is not surprising. In general, unless fires are severe, most shrub-tussock tundra types re-vegetate rapidly (Racine *et al*. 1987; Racine *et al*. 2004), even within weeks following the fire. Therefore, similar to other grassland types (Zhu *et al*. 2006), tundra fires are likely to show less difference in dNBR one year post-fire than sites

Table 8. ANCOVA comparing regressions of CBI and dNBR when grouped by pre-fire vegetation types

Source	Sum of squares (Type III)	d.f.	Mean square	*F*-statistic	*P* value
Corrected model	100.524[A]	4	25.131	170.118	<0.0001
Intercept	17.760	1	17.760	120.221	<0.0001
dNBR (covariate)	85.285	1	85.285	577.316	<0.0001
Vegetation type (fixed)	2.842	3	0.947	6.413	<0.0001
Error	41.363	280	0.148		
Total	695.055	285			
Corrected total	147.887	284			

[A]Adjusted $R^2 = 0.704$.

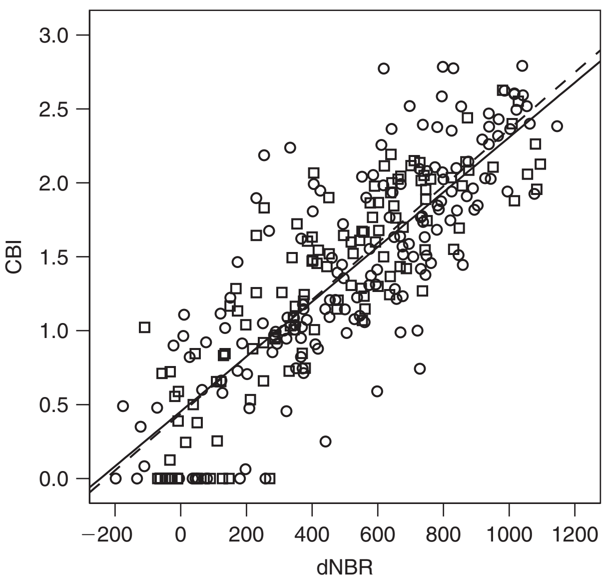

Fig. 6. Relationship of dNBR and CBI by type of assessment (initial or extended). The solid line shows the regression line for initial assessment plots, the dashed line shows the regression for the extended assessment. Initial assessment linear regression shown: CBI = 0.00185 × dNBR + 0.4539 ($F = 300.96$, $R^2 = 0.65$, $P < 0.0001$). Extended assessment linear regression shown: CBI = 0.00192 × dNBR + 0.4394 ($F = 366.44$, $R^2 = 0.74$, $P < 0.0001$). There was no significant difference between the two types of assessments.

with large trees or major structural or vegetation changes in the same amount of time.

When we pooled all the tundra plots together (irrespective of fire) it appeared that the initial assessment of tundra plots had a lower R^2 value than the extended assessment (0.44 *v.* 0.81). However, the poor fit in the initial assessment tundra plots was largely driven by six plots from the Yukon–Charley B248 fire. When these six plots were removed, the R^2 for the initial assessment of tundra plots improved to 0.78 (Fire B001, Table 4). The six low shrub/tussock tundra sites from B248 appeared to be the outliers to the overall dNBR–CBI relationship within this fire. For almost all of these plots, we had lower ground severity scores than expected for the dNBR values. Epting *et al.* (2005) discussed the B248 fire in Yukon–Charley and attributed the poor

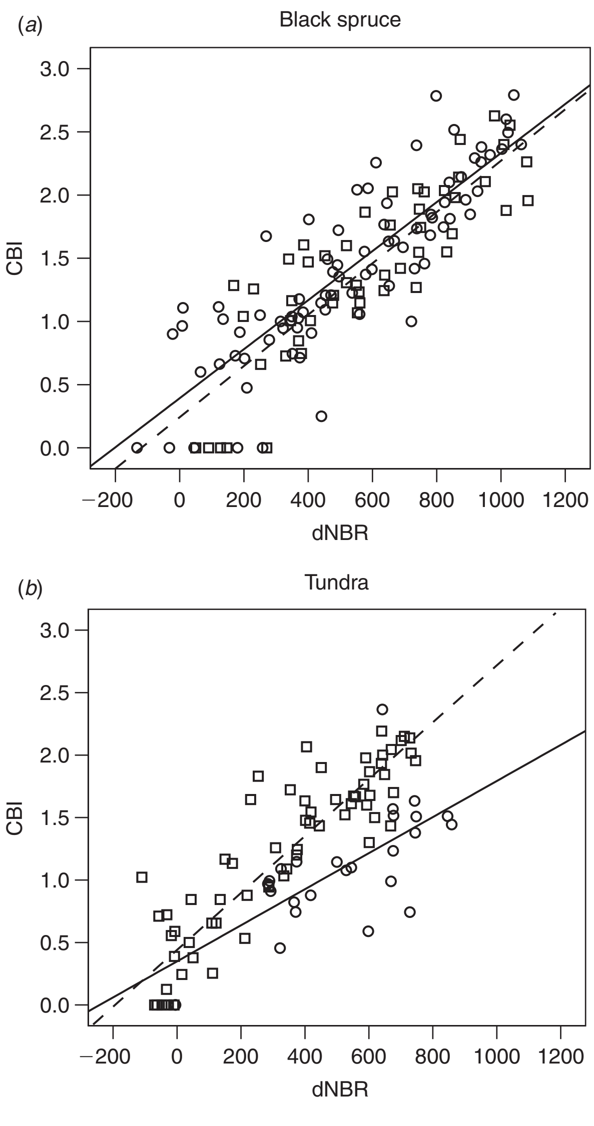

Fig. 7. The relationship between CBI and dNBR grouped by pre-fire vegetation and assessment types. Black spruce plots are shown in the top graph, tundra plots in the bottom graph. Initial assessment plots are shown with circles and the fit line is solid. Extended assessment plots are shown with squares and the fit line is dashed. A significant difference in the slopes was detected among assessment types for tundra plots.

Table 9. Results of linear regressions for CBI as a factor of dNBR when plots are pooled by pre-fire vegetation types and assessment type

Assessment type	Pre-fire vegetation type	Slope	*y*-intercept	Adjusted R^2	*F*-statistic (d.f.)	*P* value
Initial	Black spruce	0.0019	0.393	0.723	212.36 (1,80)	<0.0001
Extended	Black spruce	0.0020	0.242	0.731	139.56 (1,50)	<0.0001
Initial	Tundra	0.0014	0.349	0.429	19.02 (1,23)	0.0002
Extended	Tundra	0.0022	0.441	0.805	276.93 (1,66)	<0.0001

fit to problems with mapping severity in sites with low shrub and herbaceous types, particularly in relation to the late scene dates of the imagery (mid-September). In addition, because we collected our ground data two years post-fire in Yukon–Charley, it may have been more difficult to determine the impacts of fire in rapidly regenerating tundra sites.

Burn severity within northern regions is often addressed in terms of surface organic matter consumption, particularly within boreal forests (e.g. Viereck 1983; Kasischke and Johnstone 2005; Johnstone and Chapin 2006). The reduction of soil organic materials influences plant reproductive capabilities (Zasada 1986; Johnstone and Kasischke 2005; Johnstone and Chapin 2006), carbon emissions (Kasischke *et al*. 2000) and permafrost (Yoshikawa *et al*. 2002). We found that substrate severity scores were highly correlated with dNBR within our study. This was surprising, since it has been suggested that remotely sensed imagery is more likely to detect fire effects on tree crowns than on the ground (Hudak *et al*. 2004). The open-woodland canopies of black spruce probably allowed for detection of ground impacts by the satellite sensor and in addition over one-third of our plots were in treeless tundra. The utility of detecting organic material consumption using 30-m resolution Landsat imagery depends on the scale of application. The scale of dNBR values is likely to be too coarse to predict regeneration strategies for plants or impacts on permafrost at a plot level, particularly with regard to the variability and patchiness of consumption within a given area (Dyrness and Norum 1983). However, it may be useful in predicting broad scale vegetation patch mosaic changes, detecting potential erosion areas or estimating carbon emissions.

Within our study, we detected a significant difference in the CBI–dNBR relationship among the fires. The variability can be partially attributed to the differences in vegetation and possibly assessment type, but overall we found that the data fit best when analysed by individual fire. This raises the question of whether a single equation can be used to define severity to allow for comparisons of fires across time and space. We considered two normalisation methods: the relative dNBR (RdNBR) and calibrating the dNBR by subtracting the mean unburned bias (Zhu *et al*. 2006; Miller and Thode 2007). Miller and Thode (2007) assessed a technique called the RdNBR, which measures burn severity relative to the amount of pre-fire vegetation available. In areas of the Sierra Nevada with heterogeneous vegetation cover, Miller and Thode (2007) found that the RdNBR improved classification accuracies of the high burn severity category. In a regional analysis of our data, the RdNBR transformation did not appear to improve our results in comparison with the dNBR (Zhu *et al*. 2006). Although we had varying vegetation types, we did not have the sparse vegetation types in which the advantages of the RdNBR approach are more apparent. We also assessed normalising the dNBR by subtracting the mean unburned bias. We found that subtracting the mean unburned bias in each assessment did not change the R^2 values for the linear regressions by fire, but did alter the *y*-intercepts for each fire (J. L. Allen and B. Sorbel, unpubl. data). However, it did not appear to adequately normalise the data and significant differences were still detected among the fires. While there may be advantages to using these approaches for comparison of multiple assessments, we found that neither method removed the variability found amongst the relationships of the fires.

Conclusions

In our study, we found that burn severity mapping using Landsat TM/ETM+ imagery captured the heterogeneous nature of fire and offers a more complete description and quantification of fire's effect on the landscape. Currently, such burn severity maps are used to refine and improve final fire perimeters by fire management. In addition, burn severity maps provide a means to identify unburned islands within a burned area, which can be used to determine whether study sites within a fire perimeter have burned and to what degree of impact. In fire prone areas, burn severity data products provide a means to update fuels and vegetation GIS datasets that, without intervention, are made less accurate and relevant with each passing fire season. Severity data may also be used as a key explanatory variable for vegetation, wildlife, water quality or permafrost monitoring or research.

Within our study we detected some interesting trends regarding the variation of burn severity among the four vegetation types sampled; however, we feel that further research is warranted in the area of burn severity and fire behaviour in white spruce, deciduous forests, and tundra vegetation types. Our comparisons of initial and extended assessments were largely based on pooled data. Fire B288 was the only instance where both assessment types could be compared directly on the same fire. Relying largely on pooled data for comparison of the assessment types may have masked differences among the assessment types (Fig. 6). Although we feel there is a difference in tundra fires between initial and extended assessment, a more direct test would be to compare fires mapped by both assessment types.

Our results indicated that dNBR values were strongly correlated with field measurements of burn severity using the CBI. However, this contrasts with recent findings in Alaskan boreal forests (Hoy *et al*. 2008; Murphy *et al*. 2008). One difference among these studies is that we sampled over several years and over a wide range of severity, unlike the other studies that sampled mostly high severity sites from fires that burned during the record breaking fire season of 2004. The dNBR may have problems differentiating between severity levels at the high end of the burn severity range, as we found in our white spruce sites. We recognise there are limitations to using dNBR, and the inability to distinguish high severity areas could be problematic. However, at this time there does not appear to be a clear explanation for the overall discrepancy in results between studies of burn severity in Alaska. Based upon our results, we feel that the dNBR approach offers a prescription that can be used to meet many of the currently existing fire and resource management needs for a reliable form of burn severity mapping in Alaska.

Acknowledgements

Thanks to the staff of the National Park Service Eastern and Western Alaska Area Fire Management Programs for their assistance in collecting burn severity plot information. Special thanks go to Marsha Henderson, Larry Weddle, and Ken Adkisson for their valuable assistance with project planning and field work logistics. We also thank the Doyon Native Corporation for granting access to land within the boundary of Yukon–Charley Rivers National Preserve and the State of Alaska Department of Natural Resources for granting access to land burned by the Milepost 85 fire. Thanks to Karen Murphy with the US Fish and Wildlife Service and Jennifer Hrobak with the Bureau of Land Management for their assistance in fieldwork on the Milepost 85 fire. The authors thank the staff at the USGS EROS Data Center, particularly Stephen Howard, Don Ohlen, and Kari Pabst, for completing all scene acquisition and image processing work. Also, thanks to Carl Key and Nate Benson for all their assistance. Finally, the authors wish to recognise Brad Cella for his leadership, guidance and constant support.

References

Bureau of Land Management (2006) Alaska large fire history database (AKFIREHIST). Available at http://agdc.usgs.gov/data/blm/fire/index.html [Verified 8 July 2008]

Cocke AE, Fulé PZ, Crouse JE (2005) Comparison of burn severity assessments using Differenced Normalized Burn Ratio and ground data. *International Journal of Wildland Fire* **14**, 189–198. doi:10.1071/WF04010

Cumming SG (2001) Forest type and wildfire in the Alberta boreal mixedwood: What do fires burn? *Ecological Applications* **11**, 97–110. doi:10.1890/1051-0761(2001)011[0097:FTAWIT]2.0.CO;2

Dissing D, Verbyla D (2003) Spatial patterns of lightning strikes in interior Alaska and other relations to elevation and vegetation. *Canadian Journal of Forest Research* **33**, 770–782. doi:10.1139/X02-214

Duffy PA, Epting J, Graham JM, Rupp TS, McGuire AD (2007) Analysis of Alaskan burn severity patterns using remotely sensed data. *International Journal of Wildland Fire* **16**, 277–284. doi:10.1071/WF06034

Dyrness CT, Norum RA (1983) The effects of experimental fires on black spruce forest floors in interior Alaska. *Canadian Journal of Forest Research* **13**, 879–893. doi:10.1139/X83-118

Epting J, Verbyla D, Sorbel B (2005) Evaluation of remotely sensed indices for assessing burn severity in interior Alaska using Landsat TM and ETM+. *Remote Sensing of Environment* **96**, 328–339. doi:10.1016/J.RSE.2005.03.002

French NHF, Kasischke ES, Hall RJ, Murphy KA, Verbyla DL, Hoy EE, Allen JL (2008) Using Landsat data to assess fire and burn severity in the North American boreal forest region: an overview and summary of results. *International Journal of Wildland Fire* **17**, 443–462. doi:10.1071/WF08007

Hely C, Bergeron Y, Flannigan MD (2000) Effects of stand composition on fire hazard in mixed-wood Canadian boreal forest. *Journal of Vegetation Science* **11**, 813–824. doi:10.2307/3236551

Hoy EE, French NHF, Turetsky MR, Trigg SN, Kasischke ES (2008) Evaluating the potential of Landsat TM/ETM+ imagery for assessing fire severity in Alaskan black spruce forests. *International Journal of Wildland Fire* **17**, 500–514. doi:10.1071/WF08107

Hudak AT, Robichaud PR, Evans JB, Clark J, Lannom K, Morgan P, Stone C (2004) Field validation of Burned Area Reflectance Classification (BARC) products for post fire assessment. In 'Remote sensing for field users: proceedings of the 10th Forest Service Remote Sensing Application Conference', 5–9 April 2004, Salt Lake City, UT. On CD-ROM. (American Society for Photogrammetry and Remote Sensing: Bethesda, MD)

Johnson EA (1992) 'Fire and vegetation dynamics: studies from North American boreal forest.' (Cambridge University Press: New York)

Johnson EA, Miyanishi K (Eds) (2001) 'Forest fires behavior and ecological effects.' (Academic Press: San Diego, CA)

Johnstone JF, Chapin FS, III (2006) Effects of soil burn severity on post-fire tree recruitment in boreal forest. *Ecosystems* **9**, 14–31. doi:10.1007/S10021-004-0042-X

Johnstone JF, Kasischke ES (2005) Stand-levels effects of soil burn severity on postfire regeneration in a recently burned black spruce forest. *Canadian Journal of Forest Research* **35**, 2151–2161. doi:10.1139/X05-087

Kasischke ES, Johnstone JF (2005) Variation in postfire organic layer thickness in a black spruce forest complex in interior Alaska and its effects on soil temperature and moisture. *Canadian Journal of Forest Research* **35**, 2164–2177. doi:10.1139/X05-159

Kasischke ES, O'Neill KP, French NHF, Bourgeau-Chavez LL (2000) Controls on patterns of biomass burning in Alaskan boreal forest. In 'Fire, climate change, and carbon cycling in the North American boreal forest'. (Eds ES Kasischke, BJ Stocks) pp. 173–196. (Springer: New York)

Kasischke ES, Williams D, Barry D (2002) Analysis of the patterns of large fires in the boreal forest region of Alaska. *International Journal of Wildland Fire* **11**, 131–144. doi:10.1071/WF02023

Kasischke ES, Rupp TS, Verbyla DL (2006) Fire trends in the Alaskan boreal forest. In 'Alaska's Changing Boreal Forest'. (Eds FS Chapin III, MW Oswood, K Van Cleve, LA Viereck, DL Verbyla) pp. 285–301. (Oxford University Press: New York)

Key CH (2006) Ecological and sampling constraints on defining landscape fire severity. *Fire Ecology* **2**(2), 34–59. Available at http://www.fireecology.net/pdfs/vol2iss2/Key.pdf [Verified 8 July 2008]

Key CH, Benson NC (2006) Landscape Assessment (LA). In 'FIREMON: Fire Effects Monitoring and Inventory System'. (Eds DC Lutes, RE Keane, JF Carati, CH Key, NC Benson, LJ Gangi) USDA Forest Service, Rocky Mountains Research Station General Technical Report RMRS-GTR-164-CD. p. LA-1–55. (Fort Collins, CO)

Lentile LB, Holden ZA, Smith AMS, Falkowski MJ, Hudak AT, Morgan P, Lewis SA, Gessler PE, Benson NC (2006) Remote sensing techniques to assess active fire characteristics and post-fire effects. *International Journal of Wildland Fire* **15**, 319–345. doi:10.1071/WF05097

Miller JD, Thode AE (2007) Quantifying burn severity in a heterogeneous landscape with a relative version of delta Normalized Burn Ratio (dNBR). *Remote Sensing of Environment* **109**, 66–80. doi:10.1016/J.RSE.2006.12.006

Miller JD, Yool SR (2002) Mapping forest post-fire canopy consumption in several overstory types using multi-temporal Landsat TM and ETM data. *Remote Sensing of Environment* **82**, 481–496. doi:10.1016/S0034-4257(02)00071-8

Murphy KA, Reynolds JH, Koltun JM (2008) Evaluating the ability of the differenced Normalized Burn Ratio (dNBR) to predict ecologically significant burn severity in Alaskan boreal forests. *International Journal of Wildland Fire* **17**, 490–499. doi:10.1071/WF08050

National Interagency Coordination Center (2007) Fire Information – Wildland Fire Statistics. Available at http://www.nifc.gov/fire_info/lightning_human_fires.html [Verified 8 July 2008]

Racine CH, Johnson LA, Viereck LA (1987) Patterns of vegetation recovery after tundra fires in northwestern Alaska, USA. *Arctic and Alpine Research* **19**, 461–469. doi:10.2307/1551412

Racine C, Jandt R, Meyers C, Dennis J (2004) Tundra fire and vegetation change along a hillslope on the Seward Peninsula, Alaska, USA. *Arctic and Alpine Research* **36**, 1–10. doi:10.1657/1523-0430(2004)036[0001:TFAVCA]2.0.CO;2

Roy DP, Boschetti L, Trigg SN (2006) Remote Sensing of Fire Severity: Assessing the Performance of the Normalized Burn Ratio. *IEEE Geoscience and Remote Sensing Letters* **3**(1), 112–116. doi:10.1109/LGRS.2005.858485

Van Wagner CE (1983) Fire behavior in northern conifer forests and shrublands. In 'The role of fire in northern circumpolar ecosystems'. (Eds RW Wein, DA MacLean) pp. 65–80. (Wiley: Chichester, UK)

van Wagtendonk JW, Root RR, Key CH (2004) Comparison of AVIRIS and Landsat ETM+ detection capabilities for burn severity. *Remote Sensing of Environment* **92**, 397–408. doi:10.1016/J.RSE.2003.12.015

Viereck LA (1983) The effects of fire in a black spruce ecosystem of Alaska and Northern Canada. In 'The role of fire in northern circumpolar ecosystems'. (Eds RW Wein, DA MacLean) pp. 201–220. (Wiley: Chichester, UK)

Viereck LA, Dyrness CT, Batten AR, Wenzlick KJ (1992) The Alaska vegetation classification. USDA Forest Service, Pacific Northwest Research Station, General Technical Report PNW-GTR-286. (Portland, OR)

Wang GG (2002) Fire severity in relation to canopy composition within burned boreal mixedwood stands. *Forest Ecology and Management* **163**, 85–92. doi:10.1016/S0378-1127(01)00529-1

Yoshikawa K, Bolton WR, Romanovsky VE, Fukuda M, Hinzman LD (2002) Impacts of wildfire on the permafrost in the boreal forests of interior Alaska. *Journal of Geophysical Research* **108**(NO. D1), 8148. doi:10.1029/2001JD000438

Zasada J (1986) Natural regeneration of trees and tall shrubs on forest sites in interior Alaska. In 'Forest ecosystems in the Alaska taiga: a synthesis of structure and function'. (Eds K Van Cleve, FS Chapin, PW Flanagan, LA Viereck, CT Dyrness) pp. 44–73. (Springer Verlag: New York)

Zhu Z, Key C, Ohlen D, Benson N (2006) Evaluate Sensitivities of Burn-Severity Mapping Algorithms for Different Ecosystems and Fire Histories in the United States: Final Report to the Joint Fire Science Program. Available at http://jfsp.nifc.gov/projects/01-1-4-12/01-1-4-12_Final_Report.pdf [Verified 8 July 2008]

Manuscript received 26 February 2008, accepted 17 March 2008